Landscape Ecology **17:** 133–155, 2002.

Recent forest cover type transitions and landscape structural changes in northeast Minnesota, USA

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Received 1 January 2001; accepted in revised form 7 September 2001

Key words: Change detection, Forest classification, Forest dynamics, Forest fragmentation, Landsat, Landscape indices, Landscape pattern

Abstract

Landsat TM satellite data covering an approximate 5-year interval (1990–1995) were used to quantify spatial pattern and transition rates between forest ecological states for a 2.76 million ha region in northeast Minnesota. Changes in forest cover were stratified by Ecological Subsection, management status, and by ownership categories using a 1995 digital ownership layer. Approximately 4.2% of the 1990 mature forested area was converted to early successional types by 1995. Of this 4.2%, private lands accounted for 33%, federal lands 31%, county lands 20% and state lands 16%. Notable conversion percentages by cover type category were spruce-fir (−5.3%), aspen-birch (−4.7%), jack pine (−4.6%) and black spruce (−3.0%). Transition rates were also adjusted to fit ten-year time intervals. Shannon-Weaver Eveness and edge density of cover types increased over the study period as relative contagion and interior forest area decreased. These trends suggest both smaller patches and a more even distribution of cover types. Area of upland conifers, lowland conifers and lowland hardwoods decreased while the area of mature upland hardwoods increased in most patch size classes except the > 500 ha class which showed a substantial decrease in area. The area of early successional types increased in most patch size classes. Non-industrial private forestland had the lowest proportion of interior forest of all ownership categories – decreasing by 13.5% in five years. Smaller average cut-unit sizes and uncoordinated forest management is the likely cause since cutting rates between private and public forestland were similar.

Introduction

Minnesota is expected to increase timber harvesting by roughly 25% in coming decades to fuel the fiber demands of the paper industry (Jaakko 1994). Unfortunately, one of the problems in monitoring landscape change in this region is that the current rate of forest cutting is faster than the forests can be re-inventoried on the ground (Czaplewski 1999). For example, the ground-based U.S. Forest Service's Forest Inventory and Analysis (FIA) program is repeated only once every 10–15 years (Czaplewski 1999). Consequently, over one half of the latest FIA database is out of date at any one time, and FIA data greater than five years old are, therefore, not regarded as reliable (American Forest Council 1992). Even if the FIA database was current, these data are made up of point samples and are not well suited for studying spatially explicit forest change dynamics on a landscape basis.

One solution to the inventory problem is the use of space-borne sensors such as Landsat Thematic Mapper (TM). TM data have been found to be an efficient tool for the detection of forest canopy changes due to logging operations (Olsson 1994), insect infestation (Vogelmann and Rock 1989), acid rain (Vogelmann and Rock 1988) and forest succession (Hall et al. 1991). Our objectives were to determine if we could detect changes in forested landscape structure using Landsat TM data. We hypothesized that changes in forest spatial structure over time will vary

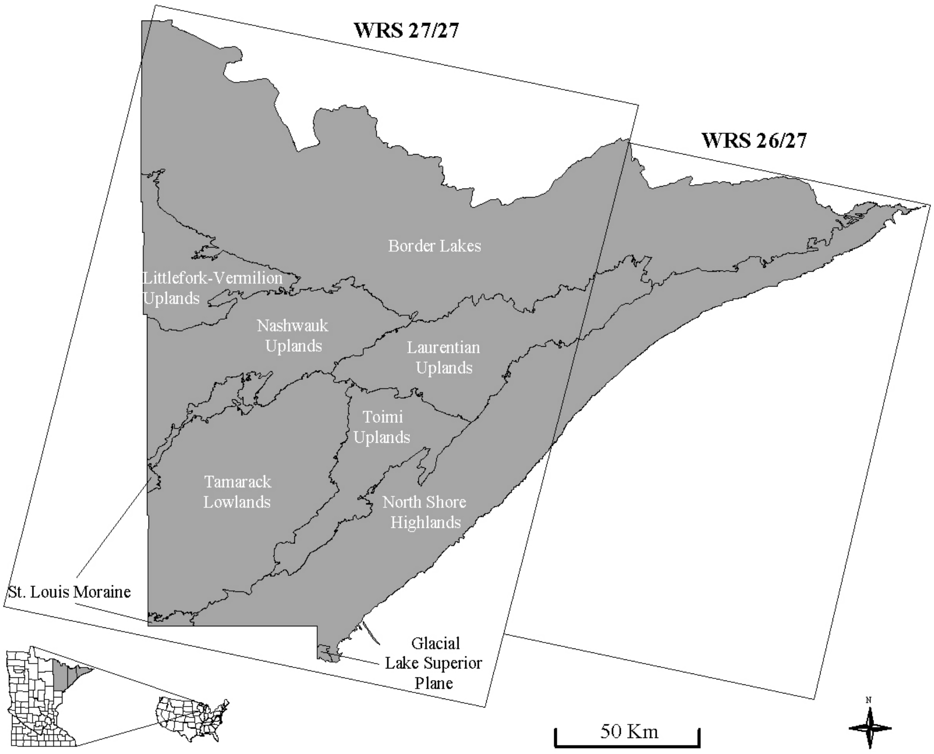

Figure 1. Study area in northeast Minnesota, two Landsat TM footprints that cover the area, and Ecological Subsection boundaries.

by ownership, forest management status, and ecological subsection.

Study area

The 2.76 million hectare study region in northeastern Minnesota is composed largely of boreal forests and includes Voyageurs National Park, Superior National Forest, and the protected wilderness of the Boundary Waters Canoe Area (BWCA) (Figure 1). Forest management practices within this region range from intensively managed private industrial forests to the unmanaged BWCA and Voyageurs National Park. It should be noted that logging has not been permitted in the BWCA since 1978. The BWCA historical disturbance regime, driven mainly by large fires (Heinselman 1973, 1996) and wind throw (Frelich and Carlton 2000), is vastly different from the disturbance regime caused by timber harvesting outside wilderness areas (Hall et al. 1991). Heinselman (1973); Hall et al. (1991); Pastor and Mladenoff (1992); Pastor et al. (1993); Frelich and Reich (1995) have previously described the ecology of this study region.

Satellite data

Two satellite footprints were needed to cover the study area: Worldwide Reference System (WRS) path/row 27/27 and 26/27 (Figure 1). TM data from 12 May 1990, 31 July 1990, 24 February 1991, 20 September 1991 and 29 July 1995 were used for path/row 27/27. Path/row 26/27 TM data included 4 March 1984, 26 May 1986, 2 July 1988, 26 September 1990 and 1 June 1994.

Summer TM data from 31 July 1990 (WRS 27/27) and 2 July 1988 (WRS 26/27) (hereafter nominally referred to as 1990) were calibrated to reflectance according to Price (1987) and geometrically registered to UTM zone 15 (NAD83) coordinates. We achieved a root-mean-squared (RMS) error of unit weight of approximately 0.37 pixels (28.5 meter pixels) for the fit between digital Landsat TM data and the 1:24,000 scale USGS topographic maps using 39 evenly distributed ground control points.

Each of the remaining scenes was calibrated to reflectance then radiometrically rectified (Hall et al. 1991) to the July TM data for each respective footprint. These data were then coregistered to the July

data. For better spatial continuity of the TM data through time, a coregistration algorithm which assigns output pixel locations based on interpolating input brightness values in two orthogonal directions (bilinear interpolation) was used. Nearest neighbor image registration algorithms preserve precise input pixel spectral information but tend to corrupt linear features due to poor spatial relocation of pixels, which may contribute a significant amount of error in multi-temporal studies (Logan and Strahler 1979). Assessments of the image-to-image registrations were performed independently by selecting 26 checkpoints per scene. All RMS errors were within 0.5 pixels (first order) for the fit between the July dates from 1990 and all other Landsat data sets.

Forest classification

The classification techniques employed in this study are essentially those used by Wolter et al. (1995) in northern Wisconsin, USA. Therefore, only a brief description of forest classification methods will be discussed. Forest canopies in this region are phenologically similar during mid-summer, and hence, satellite-based classification aids gleaned from winter, spring, and autumn phenology were made via comparison to July TM data.

July TM images from each footprint were classified into conifer, hardwood, mixed conifer-hardwood, and non-forest using Minnesota Department of Natural Resources (MnDNR) Phase II forest inventory data (Minnesota Department of Natural Resources 1995) and 1:40,000 color infrared (CIR) aerial photographs (May and September 1991) as ground truth. The hardwoods class was used to clip the September and May Landsat data for each footprint to classify black ash (*Fraxinus nigra*), maple-dominated (*Acer saccharum* and *A. rubra*) northern hardwoods and aspen-birch (*Populus tremuloides-Betula papyrifera*) cover types. The conifers class was used to clip the winter TM data for classifying tamarack (*Larix laracina*), a deciduous conifer. Winter TM data were also used to detect the presence of understory conifer components among types classified as hardwood using summer TM data.

Peak maple senescence in this region typically occurs near the autumnal equinox (Eder 1989) while black ash trees are completely defoliated by this time (Wolter et al. 1995). Conversely, aspen-birch is the first hardwood cover type to flush leaves in the spring – usually by mid-May (Ahlgren 1957). Senescent, maple-dominated, northern hardwood cover types were classified using a combination of July and September TM data. Black ash, aspen-birch and tamarack types were classified by applying thresholds to normalized difference vegetation index (NDVI) difference images: September-July, May-July and winter-July respectively. NDVI is defined as:

$$((TM4 - TM3)/(TM4 + TM3) + 1)\times 100 \qquad (1)$$

MnDNR Phase II forest inventory and CIR aerial photos were used to guide threshold selection for the difference images. Remaining cover types were classified using traditional iterative classification techniques using Phase II forest inventory, CIR aerial photos and field visits for validation (see Wolter et al. (1995)).

Overall classification accuracy for the WRS 27/27 footprint was 75% for 37 classes (KHAT 0.74) (Appendix 1). The error matrix for the WRS 26/27 footprint (not presented) was similar to the 27/27 footprint with an overall accuracy of 73% (KHAT 0.72). Accuracy assessment methodologies for both footprints were performed according to Wolter et al. (1995) where MnDNR Phase II forest inventory and 1995 CIR aerial photos were used as ground truth.

Change detection

TM scenes 7/29/1995 and 6/1/1994 (hereafter nominally referred to as 1995) represented the second time-step for footprints WRS 27/27 and 26/27 respectively (Figure 1). To detect land-cover changes in 1995, we first identified pixels that exhibited a difference, either positive or negative, in reflectance between 1990 and 1995. To accomplish this, we utilized a normalized version of the short-wave infrared/near infrared ratio (SWIR/NIR) using TM bands centered at 1.65 μm (TM Band 5; 1.55–1.75 μm) and 0.83 μm (TM Band 4; 0.76–0.90 μm) respectively. The normalized difference SWIR/NIR ratio (NDSN) is defined as:

$$((TM5 - TM4)/(TM5 + TM4) + 1)\times 100 \qquad (2)$$

This basic ratio has been used extensively in the eastern United States to detect changes in forest cover due to insect outbreaks (Vogelmann and Rock 1989) and forest decline (Vogelmann and Rock 1988). Vogel-

mann (1990) showed that the SWIR/NIR ratio is far more sensitive to forest canopy disturbance than the widely used NDVI. Interpreted CIR aerial photographs (May 1991 and September 1997) were use to determine appropriate discriminant values for change in the NDSN difference images.

After change pixels between 1990 and 1995 were identified, they were used as a template to clip the classification from 1990. The clipped 1990 classification was then used as a template to stratify and classify raw TM data from 1995 on a class-by-class basis. For example, change pixels originating from the 1990 aspen-birch class were used to focus the classification of raw TM data for 1995. In doing this, potential illogical transition classes for this time span may be ruled out in favor of more logical aspen-birch transition endpoints (e.g., upland grass/brush, regeneration, burned or flooded) when 1995 spectra are confused with illogical transition types. An aspen-birch stand that was harvested between 1990 and 1995 for example, should not be classified as ericacious brush, sphagnum moss or acid-bog conifers in 1995. This would be a highly unlikely transition of cover types within the five-year time frame. Conversely, if some of the change identified by the difference image classifies to mature hardwood it is most likely aspen-birch and can be treated as no change. Once all cover type transitions were identified, they were merged with the 1990 classification to produce a complete classification layer for 1995.

Accuracy assessment for 1995 change classes (Appendix 2) was accomplished by randomly selecting 349 reference sites from interpreted CIR aerial photographs from September 1997. Overall accuracy of the change classes was 89% (KHAT 0.87).

The spectral properties of harvested forests, and forest canopies damaged by insects can be difficult to discern using traditional classification algorithms. Qualitatively though, distinguishing between harvested forests and natural forest disturbance is relatively simple. Harvesting creates distinct geometric shapes with sharp gradients between forest and non-forest in northern Minnesota where clearcutting is the primary mode of wood removal (Plate 1). Soft, continuous gradients or highly irregular boundaries of defoliation/deforestation characterize natural disturbance. However, much of the timber that was blown down between 1990 and 1995 in the northeastern portion of this study area was harvested by salvage operations (Wayne Russ, USFS, personal comm.).

Landscape analysis

Landscape stratification

Since environment (soil, landform and climate), ownership and timber management status can contribute to spatial heterogeneity (see Crow et al. (1999)), we analyzed landscape changes by stratifying our analysis by ecological land units in the MN DNR Ecological Classification (Minnesota Department of Natural Resources 1999) and by ownership (Gap Stewardship Database, Minnesota Department of Natural Resources (1998)). Ownership categories were stratified by timber management status (managed, unmanaged and unknown). In our study, the term "unmanaged" refers to timber or forested areas in reserves, such as the BWCA, parks and natural areas, which restrict traditional forest harvesting operations. "Unknown" timber management status lands are non-industrial private forests.

We analyzed the study area overall and by subsection (Figure 1). Detailed analyses were focused on one subsection, the Border Lakes, by ownership and timber management status. This is the only subsection that contains significant amounts of mature forest cover under each timber management status category. Other subsections contain less than 5% of land that can be identified as unmanaged (Gap Stewardship Database, Minnesota Department of Natural Resources (1998)). Furthermore, the large area of wilderness (BWCA) with a natural disturbance regime within the Border Lakes Subsection serves as a landscape control for this subsection.

Transition matrix analysis

To quantify vegetation transitions we utilized the GIS overlay procedures in ERDAS Imagine 8.4 (ERDAS 1997). Cover types from a given state (time *t*) are used to clip out cover types from the next state (time *t + 1*). From these data we generated matrices of transition probabilities. Methods for calculating transition probabilities follow Pastor et al. (1993). The transition probabilities for time interval *t* were calculated by:

$$p_{i,j,t} = n_{i,j} / \sum_{j=1}^{m} n_{i,j} \tag{3}$$

where $p_{i,j,t}$ is the probability that a given area of the landscape has changed from class *i* to class *j* during

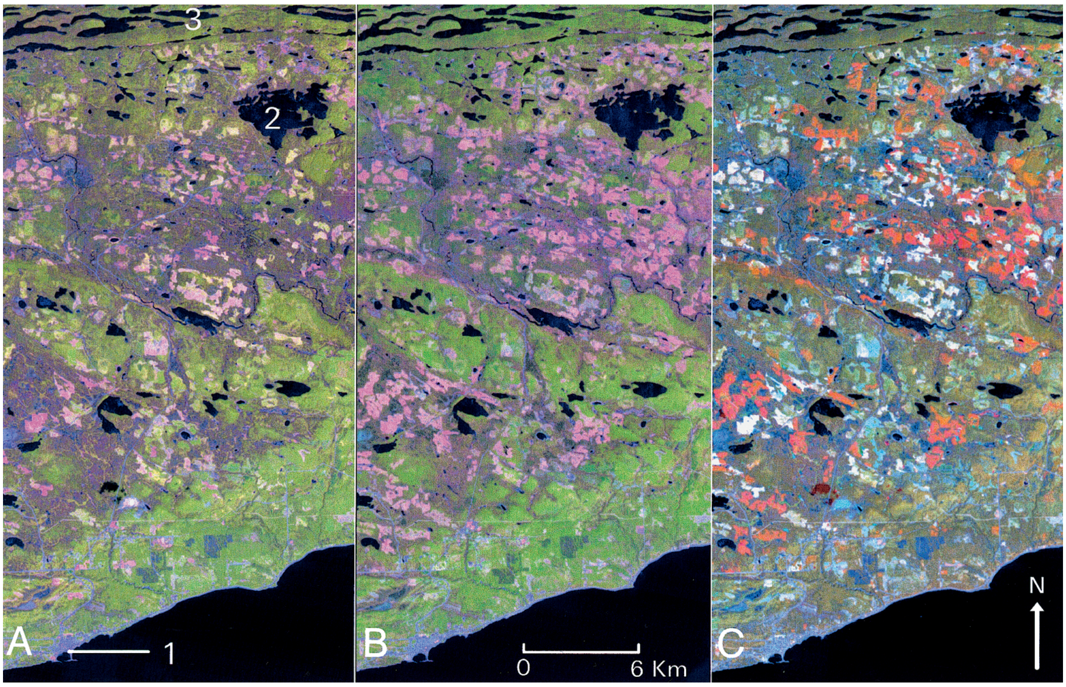

Plate 1. A. 1988 TM false color composite of a portion of the Superior National Forest and the Pat Bayle State Forest in northeast Minnesota. Image was produced using TM 1.65, TM 0.83 and TM 0.56 μm bands in the order of red, green and blue. 1 = Grand Marais, MN, 2 = Greenwood Lake and 3 = Boundary Water Canoe Area Wilderness. B. False color composite of the same area from 1994 – pink areas are forest cuts. C. 1988 to 1994 change image produced using 1994 TM 1.65, 1988 TM 1.65 and 1988 TM 0.56 μm bands in the order of red, green and blue. White areas are forest cuts from 1988 and earlier. Red areas are forest cuts that occurred between 1988 and 1994.

time interval t. $n_{i,j}$ is the land area undergoing the transition from i to j across the landscape of m classes or states. In this analysis, the sum of $p_{i,j} = 1$, indicating class i moves with some probability value to another class, including itself (Johnson and Fryer 1987). Transition probabilities over the time interval t were normalized to a 10-year interval for the periods from 1988–1994 and 1990–1995 using the equation described by Pastor et al. (1993):

$$p_{i,j} = 1 - e^{(\ln(1 - \pi))/t} \quad \text{when} \quad i \neq j \tag{4}$$

$$p_{i,i} = 1 - \sum_{j=1}^{m} p_{i,j} \tag{5}$$

Equation (4) applies to off-diagonal elements of the transition matrix. Equation (5) applies to cases when $i = j$, or on-diagonal elements and represents retention probabilities.

Landscape metrics

The software program APACK was used to calculate the following landscape statistics (Mladenoff and DeZonia 1999). To examine overall changes in landscape structure and pattern, we used two indices based on information theory. Shannon-Weaver Evenness (*SWE*) measures the evenness of the proportional distribution of patch type area. *SWE* increases with increasing evenness (McGarigal and Marks 1995).

$$SWE = \frac{H'}{\text{maximum diversity possible}} \tag{6}$$

Where measured diversity (H') =

$$-\sum_{i=1}^{\text{classes present}} P(i) \times \ln(p(i)) \tag{7}$$

and p equals the proportion of patch type i and maximum possible diversity equals ln (*classes present*). This measure describes landscape structure, but has no spatial component (Li and Reynolds 1994).

Relative contagion (*RC*) was used to examine the spatial pattern or texture of the study landscape. This index is derived from the cover type co-occurrence probabilities of the cover type adjacency matrix, which quantifies the frequency that two cover types

share a common border. RC measures the evenness of the adjacency matrix; it quantifies the degree of clumping or aggregation of cover types on the landscape. Values range between 0 and 1; high values indicate with a high degree of aggregation with fewer, large patches, while low values indicate landscapes composed of smaller patches (Li and Reynolds 1993).

$$RC = 1.0 - \left(\frac{H'}{\text{maximum possible diversity}}\right) \quad (8)$$

$$H' = -\sum_{i=1}^{n}\sum_{j=1}^{n} p_{ij}\ln(p_{ij}) \quad (9)$$

Where n is the total number of classes in a landscape, and p_{ij} is the probability of patch type i being adjacent to patch type j. Maximum possible diversity equals:

$$2 \times \ln(\text{classes present}) \quad (10)$$

To analyze finer scale changes in forest patch structure, we calculated a set of patch statistics based on patch size and size class distribution. Patch size class distribution analysis has proven useful in analyzing changes and differences in forested landscape pattern and structure (Mladenoff et al. 1993). Vegetation patch types were aggregated into six classes: upland conifer (UC), lowland conifer (LC), upland hardwood (UH), lowland hardwood (LH), regeneration (REGEN) and upland grass-brush (UG-UB). The following statistics were calculated for each class for each time period for the landscape stratification described above: number of patches, mean patch size (ha) and standard deviation, and largest patch. We analyzed patch size class distribution by calculating the number of patches and the area for the following size classes in hectares: 0–1, 1–5, 5–10, 10–50, 50–100, 100–500 and > 500.

Mean edge-to-edge patch nearest neighbor (MNN) statistics were calculated on early successional forest patches to analyze spacing differences by ownership category. Ten stratified random subsamples per category (mean subsample size ~10,000 ha) were selected to avoid fragmented ownership patterns. A one-way analysis of variance (ANOVA) on log transformed MNN data was performed to determine if mean patch spacing distances from at least one of the ownership categories was different from the rest. If so, pairwise multiple comparisons using the Tukey Test would be used to assess patch spacing differences between ownership categories.

Edge Density (ED) measures the edge length per unit area in a given landscape. The ED function that APACK uses corrects for pixel shape and saw-tooth polygon edges that are characteristic of raster data. We measured edge density for two dates for the six cover classes and total edge density for seven subsections and three management classes (managed, unknown, unmanaged) for the Border Lakes Subsection. We calculated percent difference from 1990 to 1995.

Interior Forest Area (IFA) is used as an indication of the extent of large forest patches in a landscape (Sachs et al. 1998). IFA is measured by determining the amount of forest area that is beyond a certain distance from a non-forest edge (e.g., grass, brush, roads, water). We aggregated all forest types, except regenerating forest classes, into one class then calculated IFA, according to Sachs et al. (1998), using a 100-m forest edge buffer. We calculated IFA overall and by ownership and management status categories for 1990 and 1995. We also calculated the proportion of IFA to total forest area per time step.

Results

Cover type classes from 1990 and 1995 were recoded from 37 classes to 23 classes (see Appendices 1 and 3) prior to calculation of transition matrices. The aggregation of classes both simplified analyses and elevated overall accuracy figures to 79% (KHAT 0.772) and 77% (KHAT 0.749) for WRS paths 27/27 and 26/27 respectively. Then 5-year (not presented) and 10-year transition matrices (Appendix 3) were created using output from the MATRIX program (ERDAS 1997).

Five-year transitions

Of all 1990 mature forest, trees pole size and larger, cover types combined (1.76 million ha), 1.9% of the area became native grasses, 1.8% transitioned into hardwood regeneration and 0.1% went into conifer regeneration by 1995. Transitions to flooded, assumed to be naturally induced changes, were minimal. Of the forested area in 1990, 4.2% was harvested, flooded or burned by approximately 5 years later (Plate 1, Table 1). The *percent-harvested* value was calculated by subtracting cover type transitions to flooded, emergent, water and burned from the total (Table 2). Com-

Table 1. Mature forest area changed to early successional types by ownership and timber management status category between 1990 and 1995 for the whole study area.

		1990 mature forest area (ha)	Forest changed by 1995 (ha)	1990–1995 forest change (%)	Annual rate (%)
Federal	managed	422,894	20,180	4.77	0.95
	unmanaged	307,817	2,647	0.86	0.17
State	managed	195,202	10,598	5.43	1.09
	unmanaged	57,549	1,029	1.79	0.36
County	managed	289,489	14,835	5.12	1.02
	unmanaged	3,148	34	1.08	0.22
Private Industrial	managed	37,764	3,139	8.31	1.66
	unmanaged	9,948	1,006	10.11	2.02
Non-indust. Private	unknown	415,059	19,721	4.75	0.95
Tribal	managed	16,499	456	2.76	0.55
All categories combined		1,755,367	73,645	4.20	0.84

Table 2. Percent change in mature forest area for the whole region from 1990 to 1995 by cover type and percentage of that change due to forest harvesting operations.

	% change	% of change harvested
jack pine	−4.55	92.0
red pine	−2.93	96.0
spruce-fir	−5.33	95.2
cedar	−2.34	88.2
tamarack	−2.31	81.3
black spruce	−3.04	88.1
bog conifers	−1.96	83.1
black ash	−3.19	75.7
aspen-birch	−4.72	97.6
northern hwd.	−2.72	99.4

bining human and natural disturbance, notable changes included a 5.3% decrease in the spruce-fir cover type, 4.7% decrease in the aspen-birch cover type, 4.6% decrease in the jack pine cover type and a 3.0% decrease in the black spruce cover type.

In 1990, federal lands made up the greatest proportion of mature forest area in the study area followed by private, county and state lands (Figure 2a). Of the managed federal forestlands, transitions from mature forest to non-forest and early successional forest accounted for 27.4% of the total forest disturbed (i.e., cut, flooded or burned) in the study area by 1995 (Figure 2b). State lands contributed 14.4% of the total and county lands 20.1% of the total mature forest cover disturbed by 1995. By 1995, non-industrial private lands accounted for 26.8% of the total decrease in mature forest area (Figure 2). Managed state lands, with 11.1% of the total forest area, accounted for 14.4% of the total loss of mature forest area.

Table 1 shows a breakdown by ownership and management status categories of mature forest area converted to early successional types between 1990 and 1995. Mature, federally managed forest decreased 4.8% in area. Managed state and county mature forest decreased 5.4% and 5.1% in area, respectively. By 1995, 4.8% of the mature, non-industrial private forest was disturbed due to logging, wind throw, flooding or burning. The area of mature private industrial forests, although a small percentage of the total forested area in this region, decreased by 8.3%. The ~ 10,000 ha of unmanaged, private industrial forest is primarily made up of lands dedicated to mining operations. Nevertheless, mature forest on these lands decreased by 10.1% in area by 1995.

Landscape pattern and structure

Entire study area

Shannon-Weaver Evenness (SWE) increased for all subsections from 1990 to 1995. Increases ranged from 2.3 to 4.5% (Table 3). Increasing SWE values indicate that the distribution of area by cover type is becoming more even through time. Relative contagion showed a decrease, with values ranging from −5.4 to −11% (Table 3). An overall decrease in RC suggests both smaller patches and higher evenness of cover type distribution. The North Shore Uplands subsection (Figure 1) had both the smallest SWE and largest RC values of all the subsections studied, while Littlefork-Vermillion Uplands had the highest SWE and lowest RC values. The Border Lakes, Toimi Up-

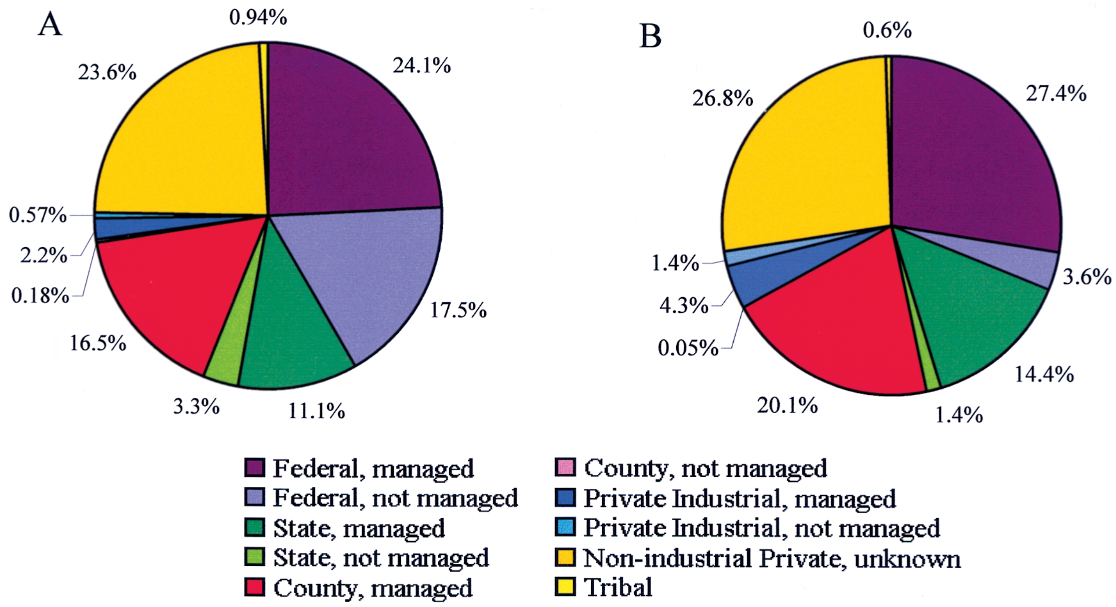

Figure 2. A. Breakdown of 1990 mature forest area (1,755,367 ha) by ownership and management status. B. Percent of mature forest area that was converted to early successional types (73,645 ha) by 1995.

lands and Laurentian Uplands subsections were all intermediate with 1995 SWE and RC values ~0.81 and 0.27 respectively. Based on SWE and RC, there appears to be three distinct groups of subsections: 1) North Shore Uplands, 2) Border Lakes-Laurentian Uplands-Toimi Uplands and 3) Nashwauk Uplands-Tamarack Lowlands-Littlefork-Vermillion Uplands (Table 3).

Border lakes subsection

Within the Border Lakes Subsection, management status showed a strong effect on SWE and RC similar to the overall study area. Managed and unknown showed similar changes with SWE increasing by 4.4 and 3.4% respectively while unmanaged showed a relatively small increase of 0.60%. RC values reveal a similar trend with changes of −11.0 and −9.3% on managed and unknown, while unmanaged decreased by 1.4% (Table 3). The BWCA wilderness contains ~91% of the unmanaged mature forest area in this ecoregion.

Edge density

Entire study area

Edge densities of upland and lowland conifer, and lowland hardwood forest decreased slightly but consistently while those of early successional types (regenerating forest and upland grass-brush) increased greatly (from 24 to 58% for regenerating forest, and from 9 to 17% for upland grass-brush). Upland hardwood forest showed the least amount of change from 1990 to 1995 (Table 4). The North Shore Uplands subsection had the lowest overall edge density (1995) of all subsections, while Tamarack Lowlands and Nashwauk Uplands were the highest. Overall subsection edge densities point to the same three distinct groups as SWE and RC (Table 4).

Border lakes subsection

Managed and unknown categories within the Border Lakes Subsection showed edge density changes similar to the overall study area with upland and lowland conifer and lowland hardwood decreasing and early successional types increasing (Table 5). Edge densities in unmanaged land, which is largely wilderness (BWCA), changed little across classes with the exception of regenerating forest, which increased by 42%, however, regenerating forest covers only 0.4% of the unmanaged landscape (Table 5).

Cover type area by patch size class

Entire study area

The area of upland conifer, lowland conifer, and lowland hardwood forest decreased for all patch size classes, with the exception of a small increase in the area of 0–1 ha in upland conifer forest (Figure 3).

Table 3. A) Shannon-Weaver Evenness (SWE) and B) Relative Contagion (RC) and percent difference (% diff) for 7 subsections for the two time periods. C) SWE and D) RC and percent difference for three timber management classes and two time periods for the Border Lakes Subsection.

A)

SWE	1990	1995	% diff
Border Lakes	0.758	0.786	3.69
Toimi Uplands	0.785	0.820	4.46
Laurentian Uplands	0.786	0.816	3.82
North Shore Uplands	0.688	0.710	3.17
Nashwauk Uplands	0.840	0.866	3.10
Tamarack Lowlands	0.855	0.875	2.34
Littlefork-Vermillion	0.870	0.897	3.10

B)

RC	1990	1995	% diff
Border Lakes	0.300	0.277	−7.67
Toimi Uplands	0.291	0.262	−9.97
Laurentian Uplands	0.294	0.268	−8.84
North Shore Uplands	0.355	0.336	−5.35
Nashwauk Uplands	0.251	0.225	−10.36
Tamarack Lowlands	0.255	0.233	−8.63
Littlefork-Vermillion	0.245	0.218	−11.02

C)

SWE	1990	1995	% diff
unmanaged	0.664	0.668	0.60
unknown	0.800	0.828	3.50
managed	0.801	0.836	4.37

D)

RC	1990	1995	% diff
unmanaged	0.361	0.356	−1.39
unknown	0.270	0.245	−9.26
managed	0.273	0.243	−10.99

Upland conifer forest showed the greatest decrease (8%) in the largest size class (> 500 ha). Thirty four percent of the upland conifer area was in the > 500 ha class in 1990. Upland hardwood forested area increased in all classes except the largest, which decreased by 13% (Figure 3a). The area of forest regeneration increased in all size classes with values ranging from 20% in the 0–1 ha class to 125% in the 50–100 ha class. The area of upland grass-brush increased by ~10% in the first 3 size classes, decreased in the 10–50 and 100–500 classes and increased by 45% in the > 500 ha class (Figure 3b).

These data show a general trend of decreasing area in most patch size classes for upland conifer, lowland conifer and lowland hardwood, while upland hardwood increased in most size classes except the > 500 ha class which showed a substantial decrease in area. The early successional types showed corresponding increases in most patch size classes.

Ten-year transition probabilities for the study area as a whole (Table 6) show that 8.8% of upland conifer, 4.0% of lowland conifer, 8.7% of upland hardwood and 4.7% of lowland hardwood classes went into either early successional forest classes or grass-brush classes. Transitions from early successional forest classes to mature forest classes were generally less than 0.5% for the ten-year period. Transitions from grass-brush classes to early successional conifer and hardwood were ~5% and 15% respectively (Appendix 3).

On federally managed forestland, 11.3% of the forest area harvested between 1990 and 1995 came from cuts > 50 ha in size (Figure 4). This figure was 17.1% on private industrial forest while non-industrial private, managed state and managed county forestland were all similar at ~8.0%. Conversely, private industrial forest had the smallest percentage (35.6%) of recently cut forest area in the < 10 ha patch size classes, while non-industrial private forest had the greatest percentage (50.5%) in these smaller size classes (Figure 4).

One-way ANOVA performed on mean nearest neighbor (MNN) distances between early successional forest patches (Figure 5) revealed a significant difference ($P < 0.001$) for one of the ownership categories. Pairwise multiple comparisons showed that unmanaged federal forest was significantly different ($P < 0.001$) from non-industrial private forest and all managed ownership categories. The other significant difference ($P = 0.011$) was between managed federal (104.7 m ± 4.5 SE) and non-industrial private forest (58.6 m ± 4.2). Unmanaged federal and state forests had the greatest distances between patches. Within the unknown and managed forest categories, private industrial had the greatest mean distance between early successional forest patches followed by federal, county, state and non-industrial private.

Border lakes subsection

An examination of the percent area of each cover type by patch size class indicates that strong differences existed between management classes in 1990 for the upland conifer class (Table 7). In the managed cat-

Table 4. Edge density (m/ha) and percent difference (%dif) 1990 to 1995 for six cover classes and 7 subsections.

	Upland conifer			Lowland conifer			Upland hardwood			Lowland hardwood			Regenerating forest			Upland grass-brush			Overall		
	1990	1995	%dif	1990	1995	%dif	1990	1995	%dif	1990	1995	%dif	1990	1995	%dif	1990	1995	%dif	1990	1995	%dif
BL	148.4	146.6	−1.2	68.7	67.8	−1.2	130.7	130.3	−0.3	9.0	8.8	−2.7	16.3	24.0	47.0	53.6	60.1	12.3	246.9	253.4	2.7
TU	78.8	75.2	−4.5	68.0	67.4	−1.0	132.9	134.3	1.0	17.7	17.3	−2.2	33.7	51.6	53.0	51.4	58.2	13.4	231.6	243.7	5.2
LU	117.3	114.7	−2.2	89.3	89.3	0.0	118.5	118.2	−0.3	13.9	13.5	−2.6	15.3	24.1	58.0	43.0	50.2	16.7	236.5	244.6	3.4
NS	59.0	57.4	−2.6	42.8	42.3	−1.2	128.7	131.0	1.8	21.8	21.5	−1.6	32.3	40.0	23.7	61.2	67.6	10.5	212.9	221.0	3.8
NU	83.5	79.6	−4.7	54.9	54.6	−0.5	142.7	143.0	0.2	13.7	13.4	−2.7	41.4	60.9	47.2	103.8	113.0	8.8	272.5	287.3	5.4
TL	63.0	60.5	−4.0	69.8	69.7	−0.2	140.3	142.1	1.3	19.8	19.7	−0.8	35.4	50.6	42.9	88.8	99.7	12.3	270.5	288.1	6.5
LM	69.4	64.9	−6.4	61.5	61.4	−0.3	129.4	128.7	−0.6	24.5	24.2	−1.5	33.3	52.4	57.5	91.3	102.0	11.7	257.0	273.5	6.4

Table 5. Edge density (m/ha) and percent difference 1990 to 1995 by timber management status and year for generalized cover classes in the Border Lakes subsection.

	UC	LC	UH	LH	REGEN	UG-UB	Overall
Managed							
1990	127.3	65.2	136.2	11.8	26.5	60.8	253.3
1995	123.2	63.5	134.9	11.4	39.1	71.2	262.0
% difference	−3.3	−2.6	−1.0	−3.5	47.8	17.1	3.5
Unmanaged							
1990	177.1	77.1	124.9	6.0	3.4	40.9	249.2
1995	177.7	77.0	125.2	6.0	4.8	43.0	252.5
% difference	0.4	0.0	0.3	−1.4	41.7	5.0	1.3
Unknown							
1990	131.7	56.3	162.0	11.2	30.9	80.5	302.7
1995	128.0	55.1	160.5	10.9	45.8	90.7	313.0
% difference	−2.8	−2.2	−0.9	−2.8	47.9	12.6	3.4

egory, 38% of the upland conifer forest was in the two largest size classes, (100–500 and > 500), in the unknown category 12% occurred in these size classes, while in the unmanaged category 72% of the upland conifer was in the largest size classes. Upland conifer area was concentrated in the smaller patch size classes for unknown management land, in the larger patch size classes on unmanaged forestland and more evenly distributed on managed forestland. For upland hardwood, managed and unmanaged showed a similar distribution, with most of the area in the largest 2 size classes, while unknown showed greatest concentration in the smaller classes.

We then compared percent difference from 1990 to 1995 for cover type area within patch size class within the Border Lakes Subsection and found that there were strong differences by management status in area by patch size class (Figure 6). Upland conifer decreased by 45 and 52% in managed and unknown in the > 500 ha class while unmanaged upland conifer showed little change across all classes. Upland hardwood increased in the smallest classes and decreased in all others in the unknown management category. Upland hardwood does not occur in the > 500 ha class in the unknown management category. In the managed category, upland hardwood had moderate

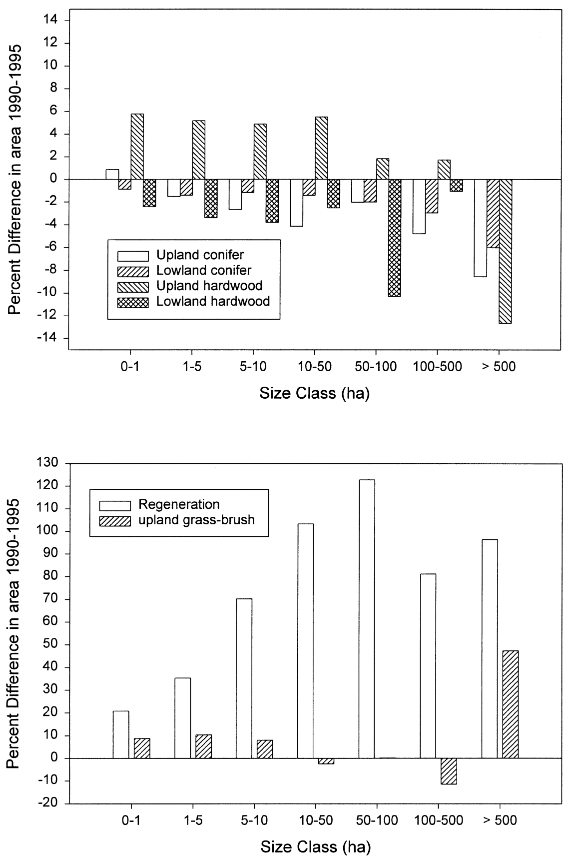

Figure 3. Percent change in area for patch size classes from 1990 to 1995 for the entire study region.

Table 6. Ten-year transition probabilities for generalized cover types for the entire study area. UC & LC are upland & lowland conifer, UH & LH are upland & lowland hardwood, GR-BR is upland grass-brush and REGEN is forest regeneration.

	1990						
		UC	LC	UH	LH	REGEN	GR-BR
1995	UC	91.25				0.13	0.37
	LC		95.96			0.05	0.21
	UH			91.33		0.33	0.37
	LH				95.34		0.03
	REGEN	2.36	0.41	5.70	2.73	98.09	5.42
	GR-BR	6.39	3.63	2.97	1.93	1.41	93.60

increases in most size classes, but decreased by 33% in the > 500 ha class (Figure 6b), which comprised 32% of upland hardwood area in 1990 (Table 7). Upland hardwood shows little change in the unmanaged category (Figure 6c). Lowland conifer had small but consistent decreases in all size classes (6% or less) where it occurs in managed and unknown categories, while in unmanaged, there was virtually no change (Figure 6a–c).

In the early successional classes, regeneration increased substantially on unknown and managed land in all size classes except the > 100 ha classes in the unknown category (Figure 6d–f). Regenerating forest made up between 4 and 5% of the managed and unknown land in 1990 (Table 8a). On unmanaged land, regeneration increased by 40% in the 0–1 ha class, in the 1–5, 5–10, and 10–50 ha classes, increases ranged from 18–22% (Figure 6f), however regeneration makes up only 0.4% of the unmanaged land (Table 8a). Upland grass-brush increased from 5 to 10% in the 3 smallest size classes in the unknown category, showed small decreases in the 10–50 and 50–100 ha classes, and increased by 25% in the 100–500 ha class. In the managed category, upland grass-brush increased in size classes where it occurred, including a 70% increase in the 100–500 ha class.

Interior forest area

Entire study area

For the entire study area, there was a 10.5% decrease in interior forest area (IFA) between 1990 and 1995 (Table 9a). On managed forestlands IFA decreased by 12.6%. IFA on unknown management status (non-industrial private) forestlands decreased by 13.5%. Tribal forestland had the greatest proportion of IFA of any management or ownership category at 53% in 1990 and 49.5% in 1995. IFA of managed county land decreased in by 15.2%, second only to the industrial categories, which decreased by 18.8 to 24.8% (Table 9a).

Border lakes subsection

Results for the Border Lakes subsection are similar to those of the full study area (Table 9b). Overall, the Border lakes subsection experienced an 8.5% decrease in IFA from 1990 to 1995. Managed forestland saw a 14.3% reduction in IFA compared to a 3.8% decrease on unmanaged forest. Unknown management status had a 13.4% decrease in IFA. The private industrial forest managed category had a 25% decrease in IFA in ~5 years, and it had the second lowest proportion of IFA in 1995 at 23.2% of mature forest area. In 1990, managed state forests had the lowest proportion of IFA of all managed forestlands at 26.8%. In 1995, managed private industrial forest had the lowest proportion of IFA (23.2%) on managed forestland. Tribal lands, though small in area by com-

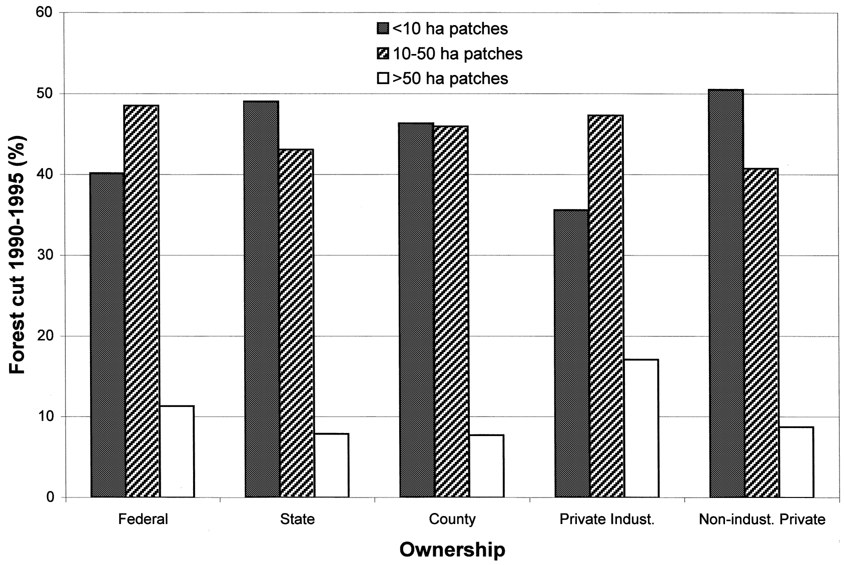

Figure 4. Percent contribution of three patch size classes to the total forest cut between 1990 and 1995 on managed forestland and non-industrial private (unknown management status) forestlands.

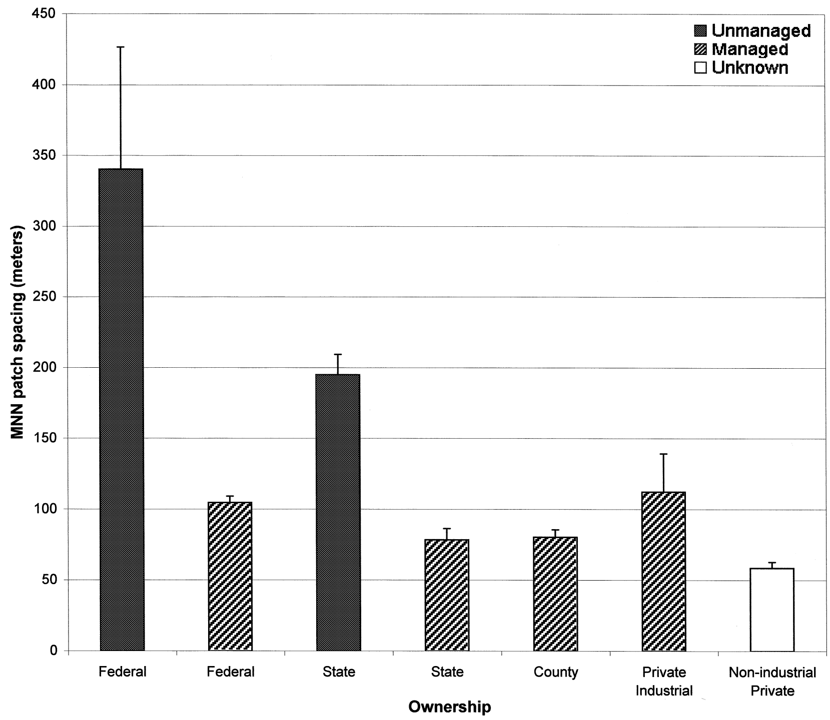

Figure 5. Mean edge-to-edge nearest neighbor (MNN) distances between early successional forest patches by ownership and management status.

parison to public ownership categories, again retained the greatest proportion of IFA at 64 and 59% respectively for 1990 and 1995. Next to tribal forestlands, unmanaged county forests had the highest proportion of IFA at 38 and 36% for 1990 and 1995 respectively. Managed county forestland had a 17.7% overall decrease in IFA in ~ 5 years – second only to managed private industrial forest (–25.5%). Next to unmanaged private industrial forest, which represented an extremely small area, unknown management status

Table 7. Proportion of cover types represented by area class categories for Border Lakes Ecological Subsection (1990).

Area (Ha)	UC	LC	UH	LH	REGEN	UG-UB
Border Lakes – Managed						
0 to 1	11.86	21.98	6.24	46.54	29.96	21.69
1 to 5	13.45	18.44	6.28	25.90	28.13	17.85
5 to 10	7.18	11.63	4.43	10.78	14.66	11.90
10 to 50	18.50	29.78	16.16	14.27	20.18	32.69
50 to 100	11.15	10.24	10.06	2.50	4.25	9.89
100 to 500	20.68	7.94	24.67	0.00	2.82	5.97
500 +	17.17	0.00	32.14	0.00	0.00	0.00
Overall	92,518	42,222	141,427	5,563	13,968	40,254
Border Lakes – Unmanaged						
0 to 1	5.76	32.63	9.23	75.87	63.07	48.29
1 to 5	5.86	24.64	8.98	15.60	21.98	22.88
5 to 10	3.25	11.94	5.28	3.16	4.85	7.73
10 to 50	9.36	22.53	17.29	5.38	10.11	13.22
50 to 100	3.45	2.92	10.17	0.00	0.00	2.26
100 to 500	12.11	5.34	23.98	0.00	0.00	5.62
500 +	60.21	0.00	25.06	0.00	0.00	0.00
Overall	172,785	44,546	124,302	2,326	1,391	19,867
Border Lakes – Unknown						
0 to 1	18.05	30.20	8.16	48.49	31.26	22.76
1 to 5	22.14	25.12	11.65	34.08	27.02	21.90
5 to 10	12.61	16.84	10.13	12.00	11.43	13.42
10 to 50	27.56	24.92	36.80	5.43	19.80	26.54
50 to 100	7.47	1.69	14.26	0.00	4.76	10.22
100 to 500	7.88	1.24	18.99	0.00	5.74	5.16
500 +	4.30	0.00	0.00	0.00	0.00	0.00
Overall	24,824	9,697	44,676	1,607	4,991	15,941

forestland (non-industrial private) had the lowest proportions of IFA per time-step of any other category (Table 9b).

Discussion

Landscape pattern and structure

The increased area of early successional forest types across the landscape has created a more even distribution of cover types in this study region as indicated by overall increases in SWE. The increased population of early successional forest patches has also fragmented large mature forest producing an overall decrease in RC (Table 3). Although there are landscape pattern differences between the subsections and groups of subsections studied, they all depict a trend of increasing SWE and decreasing RC values from 1990 to 1995 (Table 3). All forest management categories showed overall increases in ED and SWE. Reductions in ED observed within mature forest classes were offset by large increases in ED that occurred within early successional types, which produced an overall increase in ED for the study area (Tables 4 and 5). In the unmanaged forest category, where disturbance is largely due to wind-throw, fire and flooding, changes in SWE, RC and ED were slight compared with managed forest (Table 3). On managed and unknown management status lands, evenness of cover type distribution (SWE) is increasing with a corresponding decrease in forest connectivity (RC) and patch size. Logging operations between 1990 and 1995 are primarily responsible for the overall effects on SWE, RC, ED and patch size measurements observed in this study (Plate 1).

Upland conifer, lowland conifer, and lowland hardwood classes experienced decreases in ED within each major subsection studied, while the upland hardwood class showed negligible to slight increases in ED (Table 4). This is largely due to the fact that upland hardwoods (44% of 1990 classes) make up the dominant matrix of the landscape (Mladenoff et al. 1993) when compared to upland conifer (21%), lowland conifer (15%) and lowland hardwoods (2%). Currently, transitions of upland hardwood to early successional types rarely involved the complete removal of individual patches from the landscape. The upland conifer class experienced stronger negative changes in ED because transitions to early successional types had much greater effects on patch size compared to the much larger upland hardwood class. The relatively large decreases in ED reported for the upland conifers class within respective subsections is evident in Table 2 and in the ten-year transition matrix (Appendix 3) which shows that this class has experienced the largest proportional transition to early successional types (8.75%).

SWE and RC values indicate little change in cover type evenness and contagion on unmanaged forestland in the Border Lakes Subsection. On managed and unknown, percent difference values were on the high end of those recorded for all subsections within the study area indicating both increased evenness and decreased contagion which suggests continuing fragmentation of larger mature forest patches. The analysis of cover type area by patch-size class and edge density supports this interpretation.

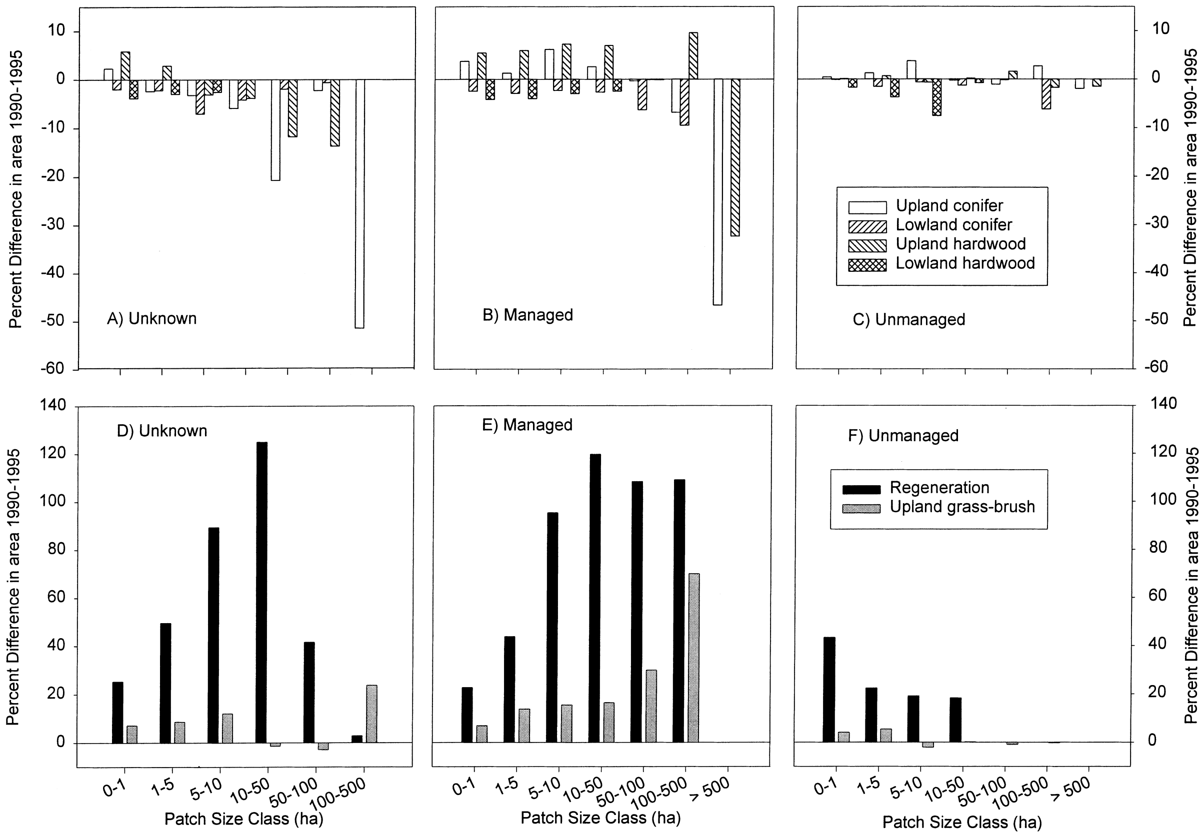

Figure 6. Percent change in area for patch size classes from 1990 to 1995 stratified by management category for the Border Lakes subsection.

Within the Border Lakes Subsection, we found that patch size class distribution differed in 1990 for upland conifer and hardwood by management status, indicating that these patterns were embedded on the landscape due to past management (Mladenoff et al. 1993; Wallin et al. 1994). In analyzing differences in patch size class distribution from 1990 to 1995, we found that managed and unknown management status lands changed in a similar way to the overall landscape, while unmanaged showed relatively little change across patch size class. In this subsection, where upland conifers are more abundant and much of the area is unmanaged (BWCA and Voyageurs National Park), the decrease in ED for the upland conifer class was substantially less than that reported for other subsections (Tables 4 and 7).

We have shown that managed and unmanaged lands here are following different trajectories; landscape pattern and structure in unmanaged remained relatively constant from 1990 to 1995, while managed lands showed increased fragmentation. Mladenoff et al. (1993) found strong differences in landscape pattern and structure when comparing a managed forest landscape to an unmanaged old-growth forest landscape in northern Wisconsin and Michigan's Upper Peninsula. Their analysis showed that on the unmanaged landscape, IFA was greater, patches were larger, landscape diversity was low, and landscape dominance high. Our comparison of landscape pattern and structure by management status in the Border Lakes showed similar results. However, the unmanaged lands here are not entirely undisturbed landscapes; portions of both the Boundary Waters Canoe Area Wilderness and Voyageurs National Park were managed for commodity production prior to receiving protected status (Heinselman 1996).

Interior forest area

Non-industrial private forestland had the lowest proportions of IFA of any ownership or management categories (Table 9a). Three factors may be responsible for this trend: 1) high rates of disturbance (clear-cutting); 2) smaller average cut-units (Li et al. 1993)

Table 8. Area and proportion of major cover types (1990) within each timber management status for A) the Border Lakes and B) the whole study region.

	Managed Area (ha)	% of total	Unmanaged Area (ha)	% of total	Unknown Area (ha)	% of total
UC	92,518	27.54	172,785	47.31	24,824	24.40
LC	42,222	12.57	44,546	12.20	9,697	9.53
UH	141,427	42.10	124,302	34.04	44,676	43.91
LH	5,563	1.66	2,326	0.64	1,607	1.58
REGEN	13,968	4.16	1,391	0.38	4,991	4.91
UG-UB	40,254	11.98	19,867	5.44	15,941	15.67
Total	335,953		365,218		101,734	
	Managed Area (ha)	% of total	Unmanaged Area (ha)	% of total	Unknown Area (ha)	% of total
UC	199,507	17.82	177,837	43.80	70,663	11.76
LC	211,836	18.93	49,824	12.27	57,044	9.50
UH	518,564	46.33	147,206	36.25	273,288	45.49
LH	29,997	2.68	3,595	0.89	14,063	2.34
REGEN	49,399	4.41	3,028	0.75	34,333	5.71
UG-UB	109,996	9.83	24,564	6.05	151,385	25.20
Total	1,119,298		406,053		600,776	

than on managed public forests (see Figures 3 and 6) and 3) a lack of spatially coordinated forest management practices on non-industrial private forestland in this region.

Spies et al. (1994) concluded that rate of forest cutting has a greater effect on edge density and IFA than the pattern in which disturbance occurs. However, other work has shown that dispersed disturbance patterns produce landscapes with greater edge density and less IFA than landscapes subjected to more aggregated disturbance patterns (Franklin and Forman 1987; Li et al. 1993; Spies et al. 1994; Wallin et al. 1994; Gustafson 1996). Timber harvest simulations on the Hoosier National Forest revealed that aggregation of cuts in space and time produced amounts of edge and IFA comparable to dispersed cutting strategies that used half the harvest rate (Gustafson 1996) indicating that differences in the configuration of harvests had greater effects on IFA than harvest rate (Gustafson and Crow 1996). When harvest rates are high and only space is considered, the advantages of aggregating harvest patches are soon lost (Gustafson and Crow 1996).

Dense cutting patterns on non-industrial private forestland (Figure 5) are compounded by the fact that they are composed of relatively small cut-unit sizes (65.7% of 1990 to 1995 cut area was from patches $\leqslant$ 16 ha). In this region, square, 16 ha ($\sim$40 acres) patches are, by far, the most common parcel size configuration of non-industrial private ownership (Gap Stewardship Database, Minnesota Department of Natural Resources (1998)). This is considerably smaller than the average managed parcel or cut-unit size (Figure 6), which is consistent with what Turner et al. (1996) found in the Olympic Peninsula and southern Appalachians; that non-industrial private lands had a greater abundance of smaller forest patches but less forest cover than public lands. Using dispersed cutting configurations, others have found that as cut-unit size decreases, large interior forest patches disappeared with increasing rapidity (Franklin and Forman 1987; Li et al. 1993). Although a remedy for negative landscape effects related to spatial configuration of ownership may be unattainable, aggregation of small cut-units into larger, but fewer, cut-units would sustain greater IFA and less edge (Gustafson and Crow 1994).

Non-industrial private forest had the greatest proportion of its cut area in smaller patch size classes (Figure 4) and the most densely arranged cut-units (Figure 5) of unknown and managed forest categories. Cut-units on federally managed forestland are more widely spaced and have a greater proportion of the area in larger patch size classes than on other managed forest categories – except private industrial forest where both of these patch metrics are greater (Figures 4 and 5). Furthermore, IFA for each time period is higher on managed federal land than on unknown

Table 9. Interior forest area by ownership, timber management status and time for (A) the overall study region and (B) the Border Lakes subsection. Also, interior forest area as a proportion of total forest area per time step, and percent change in interior forest area from 1990 to 1995. Interior forest area was calculated using a 100-meter forest edge buffer.

Ownership	Management status	Interior forest 1990 (ha)	% of 1990 forest area	Interior forest 1995 (ha)	% of 1995 forest area	% diff. 1990 to 1995
A)						
Federal	managed	139,308	32.9	123,911	30.7	−11.1
	unmanaged	109,270	35.5	105,163	34.4	−3.8
State	managed	55,294	28.3	48,483	26.2	−12.3
	unmanaged	20,866	36.3	19,740	34.9	−5.4
County	managed	78,450	27.1	66,510	24.2	−15.2
	unmanaged	1,235	39.2	1,164	37.4	−5.7
Private Industrial	managed	10,802	28.6	8,772	25.3	−18.8
	unmanaged	2,118	21.3	1,593	17.7	−24.8
Tribal	managed	8,746	53.0	8,181	49.5	−6.5
Non-industrial Private	unknown	80,579	19.4	69,729	17.6	−13.5
All Owners	managed	291,807	30.3	255,126	27.9	−12.6
	unmanaged	133,488	35.3	127,660	34.1	−4.4
	unknown	80,579	19.4	69,729	17.6	−13.5
	All combined	505,874	28.8	452,515	26.9	−10.5
B)						
Federal	managed	46,288	28.9	40,562	26.9	−12.4
	unmanaged	107,947	35.6	103,884	34.5	−3.8
State	managed	18,872	26.8	15,933	24.2	−15.6
	unmanaged	13,834	37.6	13,285	36.5	−4.0
County	managed	11,983	29.5	9,860	25.9	−17.7
	unmanaged	1,180	38.0	1,112	36.1	−5.8
Private Industrial	managed	2,302	27.4	1,715	23.2	−25.5
	unmanaged	47.3	8.7	46.8	8.6	−1.1
Tribal	managed	2,155	64.0	1,907	59.1	−11.5
Non-industrial Private	unknown	19,901	24.6	17,237	22.5	−13.4
All Owners	managed	81,272	28.8	69,669	26.3	−14.3
	unmanaged	123,009	35.8	118,327	34.7	−3.8
	unknown	19,901	24.6	17,237	22.5	−13.4
	All combined	224,181	31.7	205,232	30.0	−8.5

and other managed forestland, which likely reflects ownership-related differences in management (Turner et al. 1996). With the exception of managed private industrial forest, the rates of disturbance on managed forestland are similar across all ownership categories (Table 1). Interestingly, the disturbance rate between managed federal and non-industrial private forest is virtually identical at 0.95% per year, while IFA of the latter was the lowest of all ownership categories for both time periods. Lastly, the variability of cut-unit spacing alone may have an effect on IFA as Gustafson (1996) simulations suggested. Private industrial forest had greater variability in cut-unit spacing than managed federal forest (Figure 5), while cut-unit size remained similar. The proportion of IFA on private industrial forestland (25.3%) was similar to that on managed federal land (30.7%) even though harvest rate was ~43% greater on the former suggesting a cut-unit pattern effect on IFA.

IFA is strongly related to disturbance patch size. Large matrix forest patches (> 500 ha) have an important influence on landscape pattern and structure; they maximize forest interior (habitat) and connectivity (Mladenoff et al. (1993); Crow et al. 1999). Our analysis of patch size distribution, IFA, RC and MNN shows an overall trend toward less IFA and decreased

connectivity across the managed forest landscape in Northeastern Minnesota.

Influence of management status and physical environment on landscape pattern

Soil and landform characteristics have been shown to influence forest landscape pattern and structure in Northern Lake States forests (Pastor and Broschart (1990); Mladenoff et al. (1993); Crow et al. 1999). Crow et al. (1999) also demonstrated that the interaction of management differences related to ownership and soil-landform ecoregions influence landscape pattern. Our analysis suggests that although there is variation in landscape metrics (SWE, RC, ED) related to ecological subsection areas, management has a profound influence on landscape pattern and structure in the study area. The same shift toward increasing cover type evenness (SWE), decreasing cover type contagion (RC) and increasing edge density (ED) (especially in early successional types) occurred across all subsections. Within the Border Lakes Subsection, analyses (SWE, RC, ED, patch size) were stratified by management status and indicate pattern and structure differed between managed and unmanaged at the onset of the study. Results from 1990 to 1995 indicate continued divergence with managed and unknown showing the same shifts toward increasing fragmentation that occurred over the whole study area. Although we did not measure IFA by subsection, IFA decreased substantially in all managed forest categories, which span all subsections in the study area. IFA also varied by ownership within the managed forest category. These results suggest that the patterns imposed by management are to some degree overriding landform control and creating a more homogeneous landscape pattern across the study area. Turner et al. (1996) determined that ownership had a strong influence on both landscape pattern and change in the Olympic Peninsula and Southern Appalachian Highlands study regions. They also noted that land ownership may have important effects on ecological processes because of the strong influence of ownership on landscape pattern. Our results in northeast Minnesota also show that management, as affected by ownership strongly influences landscape pattern. Turner et al. (1996) suggest that ownership should be explicitly considered when the future health of ecosystems is assessed.

Biological diversity and forest ecosystem health

Our results show that forest patch size, interior forest area and connectivity have decreased with a concomitant increase in edge density. Additionally, current management practices may favor hardwood regeneration over conifer species. These forest landscape trends may have important implications for biological diversity and forest ecosystem health.

Landscapes with small patches, high amounts of edge and significant areas in early successional cover types likely provide benefits to valued wildlife species such as ruffed grouse and white tailed deer (Hunter 1990). However, management for high edge density and early successional cover types may have significant effects on regional biological diversity. The current high population of white-tailed deer and their browsing on eastern white pine and northern white cedar is the single most important factor in the current low regeneration rates of these two tree species (Anderson et al. 2002). The research results on the effects of edge-interior relationships on bird species populations are inconclusive for primarily forested landscapes. However, at the regional scale, increasing proportions of forest edge may compromise the viability of some edge-sensitive forest birds (Temple and Flaspohler 1998).

Changes in landscape pattern and structure may also affect forest health in this region. Recent work has shown that jack pine budworm (Kouki et al. 1997) and forest tent caterpillar (Rolands and Kaup 1995) outbreaks are more severe in highly patchy forested landscapes with high edge density and small patch sizes.

Our results suggest that current and past forest management systems may have adverse effects on biological diversity and overall forest ecosystem health in this region. However, the legacy of past forest management practices may be difficult to erase without significant reductions in disturbance rates and/or changes in minimum harvest age (Wallin et al. 1994).

Usefulness of remote sensing for monitoring landscape pattern and structure

Landsat TM data has proven to be a useful tool for mapping regional forest cover and for detecting changes in forest cover through time. Image differencing techniques using the normalized difference SWIR/NIR ratio was an efficient way to detect spec-

tral changes resulting from alterations in forest canopies.

In the past, landscape pattern and structure measurements for large landscapes were performed using coarse land cover data from one time period (Mladenoff et al. 1997). This one-time snapshot using coarse resolution data is of limited use for determining regional landscape management needs. Others have focused on detailed, photo-based land use change studies covering multiple time-steps for relatively small regions (White and Mladenoff 1994). Although these studies produce more precise pictures of landscape scale trends through time, it is uneconomical to perform such work on the scale necessary for many resource management decisions.

Satellite data affords both appropriate scale and time line for regional studies of landscape pattern and structure. One advantage to the use of remote sensing data for these inventory purposes is that one can accurately determine forest type as well as area affected and do so on a regular basis. Future effort is needed to develop a more detailed remote sensing-based time line of forest management in northern Minnesota and elsewhere. Annual observations using TM data could facilitate analysis of forest harvesting behavior in relation to variable market values of wood (Turner et al. 1996). Within the time frame we studied, the market value of aspen in this region rose from USD 7.00/cord to roughly USD 21.00/cord, and pulpwood value for many other tree species were at all time highs (Minnesota Forest Resources Council 1999). By acquiring TM data from before and after the time interval studied here, we can build a more comprehensive, species-by-species account of what has taken place in Minnesota's forests. Remote sensing technology will allow us to make more informed decisions regarding landscape-based forest resource planning and policies that affect both forests and the wildlife dependant upon them.

Lastly, it should be noted that conifer regeneration appears to be under represented in the ten-year transition matrix (Appendix 3). Based on field observations and 1984 TM data, we believe most of the younger conifer regeneration patches that originated in the late 1970s and early 1980s, as a result of cutting 1933–34 origin conifer plantations, are seen as either grass or brush using 1990 TM data. Conifer regeneration (< 15 years old) does not form closed canopy patches like hardwood regeneration patches do, which allows brush and grass in the young conifer patch matrix to dominant the TM spectral signature. When conifer patches approach crown closure, their TM spectral signature becomes more distinct, which may explain why a substantial amount of the 1990 grass and brush class moved to conifer regeneration by 1995.

Summary

We used remote sensing technology to classify forest cover to a near-species level for a 2.76×10^6 ha area and, by using image ratio differencing techniques using the normalized difference SWIR/NIR ratio (NDSN), we were able to efficiently identify changes that occurred in the forest cover during a 5-year period.

Overall, 4.2% of the mature forested area was changed in the five year period studied. Of this, 5.3% of the spruce-fir, 4.7% of the aspen-birch, 4.6% of the jack pine and 3.2% of the black spruce cover type changed to early successional types. Forest harvesting on managed forests was highest on private industrial land at ~1.7% per year and lowest on tribal land, ~0.55% per year. Managed State forests had the greatest rates of disturbance of all publicly managed forestland (~1.1% per year).

We stratified the landscape by ecological subsection, ownership and timber management status and found that landscape structure varied significantly by each of the three stratification categories. The most dramatic landscape pattern effects appear to be driven mostly by management status and ownership. Tribal forestlands had the highest percent of IFA, while non-industrial private forests had the lowest. Non-industrial private lands have a greater proportion of their cut area in smaller patch sizes than managed public or private industrial forests. Differences in IFA observed between managed public, non-industrial private and private industrial forests are most likely due to differences in cut-unit size, harvest spacing patterns and, to a lesser degree, rate of harvest, and hence, reflects the management and land-use strategy of the owner (Turner et al. 1996).

Cover type area by patch size class analyses showed a general trend of decreasing area in most patch size classes for upland conifer, lowland conifer and lowland hardwood. Upland conifer showed its greatest decrease (8%) in the largest size class (> 500 ha). Upland hardwood showed an increase in all patch size classes except the largest class, which showed a 13% decrease in area.

In Summary:

- Harvest rates vary by timber management status, but show less variability by ownership category.
- Fragmentation is greatest on non-industrial private forests. Overall fragmentation trends may influence both biological diversity and ecosystem functions.
- Patterns of ownership affect landscape patterns (non-industrial private versus managed public forests).
- Our analysis of RC, patch size distribution, IFA, and MNN shows an overall trend toward less IFA and decreased connectivity across the managed forest landscape in Northeastern Minnesota.
- Cut-unit size and arrangement appear to have a greater effect on IFA than observed rate of harvest.

Acknowledgements

We greatly appreciate comments we received on early versions of this manuscript from John Pastor, Jerry Niemi, JoAnn Hanowski, Malcom Jones and Jan Green. We also appreciate the critical comments we received from three anonymous reviewers; their input greatly improved the quality of this manuscript. Funding to prepare this paper was provided by the Minnesota State Legislature for the Environmental Trust Fund, as recommended by the Legislative Commission on Minnesota Resources in support of the Minnesota Forest Bird Diversity Initiative, and NASA grant LCLUC99-0154-006. This is contribution number 300 of the Center for Water and the Environment and number 47 of the Natural Resources Geographic Information Systems (NRGIS) Laboratory of the Natural Resources Research Institute, University of Minnesota, Duluth

Reference Data

	JP	JP-h	RP	RP-h	SF	SF-h	C	C-h	T	BS	ABC	C-rgn	BA	BA-c	BA-uc	AB	AB-c	AB-uc	NH	NH-c	NH-uc	H-trn	H-rgn	bare	water	E-aq	Em	Sphag	UG	LG	Agr	UB	LB	Eric	Dev	Road	LH	C-loden	**Row Total**	**User Error**		
JP	**96**	2	7	3	3	5		1	1	1								2						1															122	78.7	JP	jack pine
JP-h	2	**78**															1																						81	96.3	JP-h	JP/hwd. mix
RP	4		**83**	4	3	2	1			2		2					4	2																					107	77.6	RP	red pine
RP-h			1	**41**						2	2																												46	89.1	RP-h	RP/hwd. mix
SF		6	8	5	**88**	9	2	4		2	1						8	2														1	3						139	63.3	SF	spruce/fir
SF-h	2		1	5	8	**108**		1		2	2					1	9	1																					140	77.1	SF-h	SF/hwd. mix
C		2			3	1	**58**	5	1	5	8				2		2											1					4						92	63.0	C	cedar
C-h							7	**35**			1																											1	44	79.5	C-h	C/hwd. mix
T		2				2		1	**44**	4	8			1														1										2	65	67.7	T	tamarack
BS	1	7			3		1		1	**172**	8							1															2						196	87.8	BS	black spruce
ABC	1				6	2	4	6	1	15	**169**			1	1	1	4	2											1				11	2					227	74.4	ABC	acid bog conifers
C-rgn	2		2	1	1							**46**																	1			1							54	85.2	C-rgn	conifer regeneration
BA						1		1			2		**234**		7	3	1	2	2								1						4				11	3	272	86.0	BA	black ash
BA-c			1			1	1	2			4		11	**104**	3		1																5				1	2	136	76.5	BA-c	BA/con. mix
BA-uc														2	**53**	5	2																						62	85.5	BA-uc	BA/con. understory
AB	3	6	2		5	4	1		1	2			8	2	2	**242**	19	8	9							7	6		1	3		4	11		7		6	3	362	66.9	AB	aspen/birch
AB-c	10		5	9	13	10	6	5	2	3	5		10	6	3	26	**314**	15		1						7	2			8		2	18				6		486	64.6	AB-c	AB/con. mix
AB-uc			2	2		1							1	2		3	2	**98**					1						1				2				1		116	84.5	AB-uc	AB/con. understory
NH																7	3	1	**63**	2	1		2			3	3		1								1		87	72.4	NH	northern hardwoods
NH-c																	2			**19**																			21	90.5	NH-c	NH/con. mix
NH-uc						1												1			**5**																		7	71.4	NH-uc	NH/con. understory
H-trn																						**11**	2																13	84.6	H-trn	transitional hardwood
H-rgn										1							5	11	4	1		3	**274**	6		2	1	2		1	1	5	8		2		6	1	334	82.0	H-rgn	hwd. regeneration
Bare	3											2					1	1					12	**116**					1		2		1		11				150	77.3	Bare	bare ground
W								1																	**77**	3	2						4						87	88.5	water	water
E-aq																										**48**		3											51	94.1	E-aq	aquatic emergent
E																	1						1	1	1		**57**			1									62	91.9	E	emergent
Sphag																												**46**	5										51	90.2	Sphag	sphagnum moss
UG	2			2						1		2	4		2	1	7					1	10	21		1	2		**37**	4		3	6				1		107	34.6	UG	upland grass
LG										1	25	1	3				3	1					3			7	12	1		**105**	5		17	5	17				206	51.0	LG	lowland grass
Agr																															**31**					2			33	93.9	Agr	agriculture
UB				1								3	3			5	6	2					13	8		1	1	1		2	1	**39**	8					0	94	41.5	UB	upland brush
LB	1			1					2	3	11		13		3	1	5						7	1	1	7	9	1		9		2	**189**		9		5	2	282	67.0	LB	lowland brush
Eric										3	18												1			1	1						2	**29**	2				57	50.9	Eric	ericacious brush
Dev																							1							2					**228**	10			241	94.6	Dev	developed
Road	3																							1			2	1	1		1	1	1		3	**153**			167	91.6	Road	roadways
LH													2	1	1																						**35**		39	89.7	LH	other lowland hwd.
C-loden	1						1		1		1																	1										**28**	33	84.8	C-loden	low density conifers
Column Total	131	103	112	74	133	147	82	62	54	219	265	56	289	119	77	295	400	150	78	23	6	15	327	155	79	87	99	58	49	135	41	58	296	36	279	165	73	42	4869 diag. = 3653			
Producer Error	73.3	75.7	74.11	55.4	66.17	73.5	70.7	56.5	81.5	78.5	63.8	82.1	80.97	87.4	68.8	82	78.5	73.7	80.8	82.61	83.33	73.33	83.8	74.84	97.5	55.2	57.6	79.3	75.5	77.8	75.6	67.2	63.9	80.6	81.7	92.7	47.9	66.667		75.02		

Classified Data

KHAT = 0.739 Overall Accuracy = 75 %

Appendix 1. Confusion matrix for the 1990 forest classification (WRS 27/27).

Reference Data

Classified Data	con. reg.	hwd. reg.	water	emergent	grass	dom. gr.	low. gr.	upland gr.	low. br.	developed	flooded	burned	**Row Total**	**User Error**
conifer regen.	**48**				2								50	0.96
hardwood regen.	2	**52**			3			1					58	0.90
water			**26**	1									27	0.96
emergent				**12**									12	1.00
native grass	12	5			**67**	1	2		1	1			89	0.75
domestic grass					1	**13**							14	0.93
lowland grass					2	1	**17**						20	0.85
upland brush		2						**23**					25	0.92
lowland brush		1						1	**21**				23	0.91
developed										**12**			12	1.00
flooded											**15**		15	1.00
burned												**4**	4	1.00
Column Total	62	60	26	13	75	15	19	25	22	13	15	4	349	
Producer Error	0.77	0.87	1.00	0.92	0.89	0.87	0.89	0.92	0.95	0.92	1.00	1.00	Diag. total	310

Overall accuracy of change classes **88.8** **KHAT** **0.871**

Appendix 2. Accuracy assessment for 1990 to 1995 change classes.

1990

2000	JP	RP	SF	C	T	BS	ABC	C-rgn	BA	AB	NH	H-rgn	Water	Em	UG	Ag	LG	UB	LB	Dev
jack pine	**91.01**							2.18				0.01	0.06	0.14	0.17		0.40	0.10	0.01	0.01
red pine		**94.18**						3.59					0.01	0.01	0.02		0.03	0.02		0.01
spruce-fir			**89.44**					2.28				0.02	0.07	0.16	0.28	0.01	0.42	0.09	0.20	0.01
cedar				**95.35**				0.27						0.01	0.01		0.07	0.01		
tamarack					**95.39**			0.36					0.01	0.09	0.02		0.16	0.05	0.01	
black spruce						**93.96**		0.42					0.02	0.03	0.02		0.16	0.03	0.01	
bog conifer							**96.09**	2.16					0.02	0.17	0.08	0.01	0.33	0.08	0.07	0.04
conifer regen	0.05	0.09	0.17	0.02	0.03	0.01	0.05	**87.08**	0.03	0.31	0.04	1.66	0.03	0.19	2.19	0.07	0.20	2.40	0.15	0.29
black ash									**93.65**					0.03	0.04		0.12	0.01	0.01	0.00
aspen-birch										**90.66**	0.25	0.33	0.03	0.08	0.89	0.03	0.29	0.18	0.06	0.04
northern hwd											**94.60**				0.04					
hwd regen	0.85	0.76	3.60	0.36	0.69	0.11	0.46	0.11	2.67	5.54	3.26	**95.62**		0.01	11.31	1.17	0.74	2.78	0.42	0.04
water	0.29	0.14	0.31	0.27	0.30	0.24	0.25	0.02	0.62	0.13	0.02	0.36	**99.42**	1.08	0.37	0.45	1.20	0.52	0.72	0.20
emergent	0.02		0.01		0.02	0.02	0.02	0.02	0.01	0.01			0.08	**91.87**	0.02	0.01	0.12	0.06	0.05	0.02
native grass	6.28	4.26	5.51	2.84	1.14	2.47	1.03	0.05	1.02	2.74	1.60	0.70	0.11	1.38	**74.40**	0.63	2.42	2.20	0.68	0.98
domestic grass								0.02		0.01	0.04	0.16		0.06	4.42	**96.82**	0.13	0.13	0.02	
lowland grass	0.70	0.36	0.48	0.86	1.77	2.40	1.51		0.83	0.17	0.02	0.08	0.13	2.60	0.48	0.32	**90.71**	0.43	0.85	0.07
upland brush	0.24	0.08	0.11			0.05	0.01			0.10	0.02	0.38		0.02	2.61	0.08	0.04	**89.78**	0.03	0.10
lowland brush	0.05	0.01	0.07	0.01	0.05	0.19	0.07	0.03	0.06	0.03	0.01	0.10		0.52	0.31	0.20	1.87	0.08	**96.08**	0.03
developed	0.09	0.03	0.10	0.02	0.07	0.09	0.11	1.35	0.18	0.20	0.13	0.52		1.25	2.28	0.15	0.27	0.96	0.11	**98.15**
flooded	0.23	0.08	0.18	0.26	0.54	0.44	0.38	0.08	0.92	0.09		0.04	0.02	0.30	0.04	0.05	0.32	0.12	0.49	0.01
burned	0.19			0.02		0.01	0.01													

Appendix 3. Ten-year transition probabilities for the whole study region. Diagonal elements are retention frequencies and off-diagonal elements are transition frequencies.

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